INVESTIGATING THE PERFORMANCE OF A LABORATORY-SCALE ECOLOGICAL SYSTEM TO TREAT DAIRY WASTEWATER

A. M. Keppler, J. F. Martin

ABSTRACT. *The potential for an ecologically engineered treatment system to improve water quality is supported by the effective treatment of wastewater by existing systems such as Living Machines and treatment wetlands. This article describes these treatment systems and their application for waste treatment and presents preliminary results of a laboratory-scale system to treat dairy wastewater. This study used an ecological treatment system, similar to a Living Machine, consisting of five 230 L tanks, that included an anaerobic tank, two aerobic tanks, a clarifier, an aquaculture tank, and a 115 L wetland mesocosm. The preliminary results demonstrate the potential for this technology to be used in the treatment of dairy wastewater following further research and modifications. Approximately 80% of the solids added were removed in the first three tanks. Reductions in TN and TKN were as great as 90% and 94%, respectively. The reductions in NH_4-N were consistently greater than 99%, and as much as 32.3% of the NO_3-N was removed. A limited amount of PO_4-P was removed by the system, only 28.2% over 168 h of treatment. Qualitative results demonstrated the potential of these systems to utilize waste inputs to produce value-added products.*

Keywords. *Dairy wastewater, Ecological engineering, Ecological treatment systems, Living machines, Nutrient removal, Value-added products, Wastewater treatment.*

The treatment and disposal of wastewater is an important challenge for animal production facilities. Inadequate methods of treatment have degraded the water quality of streams and rivers. More stringent environmental regulations require greater reductions of nutrients and solids prior to discharge and limit field applications of manure. In response to these problems, new comprehensive methods of treatment that rely on renewable energies and that reuse waste to make value-added products are needed to protect the downstream environment and ensure economic viability. One alternative is ecologically engineered treatment systems. These systems have been successful in the treatment of municipal wastewater and may offer a more sustainable and cost-effective approach to treat wastewater from animal production facilities. Ecological treatment systems also offer the possibility of using wastewater as a resource to produce value-added products such as vegetative and fish biomass.

The potential for an ecologically engineered treatment system to improve water quality of dairy wastewater is supported by the effective treatment of various types of wastewater by previously developed ecological systems. Living Machines are one example of an ecological treatment system and represent the integration of many advances in this field (table 1). They have proven successful in municipal sewage applications by reducing levels of BOD, TSS, TKN, NH_4, NO_3, TN, and TP to advanced tertiary quality (Todd et al., 2003; Austin, 2000; Todd and Josephson, 1996; Peterson and Teal, 1996). The use of algal turf scrubbers for treating contaminated groundwater has reduced levels of phosphorus by 46% and removed COD, solids, and inorganic elements (Craggs et al., 1996; Adey et al., 1996). In addition, periphyton communities have been used in these engineered systems, resulting in 80% removal of NH_4 and 70% removal of PO_4 (Vymazal, 1988). Treatment wetlands have also been a common management tool for improving the quality of polluted waters. Their use to treat tertiary municipal wastewater as well as nonpoint-source agricultural runoff has increased during the last decade. In the treatment of dairy farm wastewater, Tanner et al. (1995) showed that treatment wetlands could produce reductions of TN of 41% to 75%. Constructed wetlands designed to treat dairy farm wastewater in Fredrick County, Maryland, resulted in significant reductions in total nitrogen (98%), ammonia (56%), BOD (97%), and TSS (96%) (Schaafsma et al., 2000).

Most wastewater treatment plants' NPDES permits do not require monitoring of P removal from the wastewater. However, in areas with discharge into scenic rivers or protected water bodies, a discharge limit of 1 mg L^{-1} or lower may be required (Metcalf and Eddy, 2003). Phosphorus removal levels have varied for existing ecological treatment systems. While substantial P removal has been achieved in ecological treatment systems with high P concentrations, it has been a challenge to reach the P concentrations required in some areas (Austin, 2000; Hamersley et al., 2001; Todd and Josephson, 1996; Peterson and Teal, 1996). A solar aquatic system produced an 87% reduction in TP in the treatment of

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Table 1. Summary of recent applications of ecological systems to treat wastewater.

Ecosystem Technology	Applications Tested	Result Summary	References
Living Machines			
Living machines	Treatment of municipal sewage	Reduced BOD, TSS, TKN, NH_3, NO_3, TN, and TP to advanced tertiary quality	Todd et al., 2003
Living machine	Treatment of municipal sewage	Reduced BOD, TSS, TKN, NH_3, NO_3, TN, and TP to advanced tertiary quality	Austin, 2000
Living machine	Treatment of municipal sewage	Reduced BOD, TSS, TKN, NH_3, NO_3, TN, and TP to advanced tertiary quality	Todd and Josephson, 1996
Living machine	Treatment of confectionary wastewater	Greater than 99% reductions in BOD, TSS, fats, oils, and grease	www.livingmachines.com
Solar aquatic system	Treatment of municipal sewage	68% nitrogen reductions	Peterson and Teal, 1996
Solar aquatic system	Treatment of municipal sewage	Reduced TN by 99%	Hamersley et al., 2001
Algae and Periphyton Systems			
Algal turf scrubbers	Treating industrial contaminated groundwaters	Removed COD, solids and inorganic elements to drinking water standards	Adey et al., 1996
Algal turf scrubbers	Treatment of municipal sewage	Reduced phosphorus by 46%	Craggs et al., 1996a
Periphyton communities	Treatment of polluted stream	Removed 80% NH_4 and 70% PO_4	Vymazal, 1988
Green-algae treatment	Treatment of polluted river	Reduction of nutrients (N and P), heavy metals, and PCBs	Sladeckova, 1994
High-rate algal ponds	Treatment of municipal sewage	Significant reduction of NH_3, COD, total P, and fecal coliform	Canovas et al., 1996
High-rate algal ponds	Treatment of tannery effluent	90% decrease in SO_3 and NH_3; reductions of PO_4 and odor	Rose et al., 1996
Controlled stream mesocosms	Treatment of municipal sewage	Treated secondary sewage to tertiary standards	Craggs et al., 1996b
Aquaculture Integrated Systems			
Aquaculture-wetland ecosystem	Treatment of municipal sewage integrated with fish production	97% reduction in nitrogen; 95% gain in fish biomass	Costa-Pierce, 1998
Periphyton and fish treatment system	Treatment of N and P enriched water and fish production	Greater N and P removal (50%) with fish, 90% gain in fish biomass	Drenner et al., 1997
Filter feeders and fish treatment system	Sequential isolation of phyto-plankton, zooplankton and fish	Better treatment and greater fish biomass with isolated components	Smith, 1993
Polyculture earthen ponds	Manure and/or fertilizer additions as a food source	Manured fish growth yield of 29.5 kg ha^{-1} d^{-1}; fertilizer fish growth yield of 27.2 kg ha^{-1} d^{-1}	Schroeder et al., 1990
Polyculture earthen ponds	Use of manure as a food source to farm fish	Fish growth rate affected by their own stocking rate not manure or other species	Wohlfarth et al., 1985
Polyculture ponds	Food selection of different fish species that are fed manure	Fish biomass gain and use of all levels of the food chain	Spataru et al., 1983
Aquaculture	Use of manure as a food source to farm fish	Fish biomass gain and reduction in feed costs	Wohlfarth and Schroeder, 1979
Treatment Wetland Systems			
Treatment wetland systems	Treatment of dairy farm wastewaters	Reductions of 41% to 75% in TN and 36% to 74% in TP	Tanner et al., 1995
Treatment wetland systems	Treatment of dairy farm wastewaters	Reductions of 98% in TN, 56% in ammonia, 96% in TP, 84% in PO_4, 96% in TSS, and 97% in BOD.	Schaafsma et al., 2000
Treatment wetland systems	Treatment of dairy farm wastewaters	Reductions of 60% in NO_3, 68% in TP, 94% in TSS, and 85% in BOD	Newman et al., 2000
Treatment wetland systems	Treatment of dairy farm wastewaters	Reductions of 61% in BOD, 43% in ON, 26% in NH_3, and 28% in PO_4.	Geary and Moore, 1999

domestic and commercial raw septage with greater initial phosphorus concentrations (48 mg L^{-1} influent TP) (Peterson and Teal, 1996). In contrast, a Living Machine treating municipal sewage with lower initial phosphorus concentrations (3.90 mg L^{-1} influent TP) reduced total phosphorus by only 50% (Todd and Josephson, 1996).

Previous studies support the integration of aquaculture with wastewater treatment. Asian cultures have a long history of supporting fish production with agricultural waste (Wohlfarth and Schroeder, 1979; Wohlfarth et al., 1983; Spataru et al., 1983; Wohlfarth et al., 1985; Schroeder et al., 1990) (table 1). Western scientists have begun to investigate the inclusion of aquaculture with waste treatment. Aquaculture systems in which waste additives were applied have supported the growth of fish (Drenner et al., 1997; Smith, 1993). In an integrated aquaculture-wetland ecosystem study (Costa-Pierce, 1998), the tertiary treatment of municipal wastewater resulted in fish biomass gains. In addition to fish production, these systems resulted in greater improvement in water quality than treatments without fish. Drenner et al. (1997) traced nitrogen and phosphorus removal from periphyton assimilation to fish biomass (*Tilapia mossambica*)

and fish feces in sediments. In a series of stabilization ponds, Smith (1993) isolated phytoplankton, zooplankton, and fish (silver carp and channel catfish) to improve water quality and increase fish production.

While ecological treatment systems have been successful in treating municipal and industrial effluents, limited research has been conducted on applying this technology to highly concentrated animal production discharges. Agricultural production facilities produce wastewater with high concentrations of nitrogen, phosphorus, and solids. An ecologically engineered treatment system could offer a solution to treating the wastewater from such a facility. This article presents preliminary results from a laboratory-scale study testing the ability of an ecologically engineered treatment system to improve water quality of dairy wastewater. This article focuses on the potential transformation and removal of pollutants to improve water quality and provide habitats for fish and vegetative communities. While animal pathogens are a concern, they have not been addressed in this preliminary study.

Methods

Design of the Ecologically Engineered System

During the spring of 2002, a laboratory-scale ecological treatment system (ETS) was designed and constructed to test the ability of these types of systems to treat highly concentrated dairy wastewater from the dairy facility at Waterman Agricultural and Natural Resources Laboratory at The Ohio State University (OSU). The waste stream was composed of manure, urine, straw, and rinse water from a tie-barn feeding stall. The treatment system consisted of five 230 L tanks, including one anaerobic tank (T1), two aerated vegetative tanks (T2 and T3), one clarifier (T4), and one aquaculture tank (T5), and a 115 L surface flow wetland mesocosm (T6), resulting in a total volume of 1136 L (fig. 1). All tanks were left open, and any gases produced during treatment were released to the atmosphere. The connections between each tank were made with 0.75 in. (1.9 cm) flexible PVC hose, except for the connection between the two vegetative tanks. This connection was made with 2 in. (5 cm) flexible PVC hose to allow for fish movement between the vegetative tanks.

Flow began with the addition of 15 L of dairy wastewater to the anaerobic tank (T1) each week. At this preliminary stage, odor control devices were not included. The initial waste load of 15 L per week was small, and the resulting odor was minimal and diminished after approximately 2 h after addition of waste.

The anaerobic tank was designed to provide the initial digestion of the wastewater and an environment for denitrification. It was an open tank design (i.e., no gas collection) and was filled with lava rocks to a height of 38 cm to increase the surface area available for microbial growth. The anaerobic tank was initially seeded with wastewater sludge from a local wastewater treatment facility as well as local wetland environments to add a source of microbes to digest solids. As the wastewater moved through the lava rocks, the microbes degraded the bioavailable carbon and removed oxygen from the water column. A 2.5 in. (6.3 cm) PVC pipe was aligned vertically in the tank to allow for the collection of water samples from the bottom of the tank. The water was pumped from the bottom of T1 in two parallel pipes using airlift pumps to the mid-elevation of the first vegetative tank (T2) (fig. 1). The flow throughout the rest of the system was by displacement. The system had one feedback loop that moved water and sediment from the bottom of the second vegetative tank (T3) to the top of T1 by means of an airlift pump at a flow rate of 3:1 (fig. 1). The retention times for each tank were calculated as 2.2 h in T1; 30 min for T2, T3, and T4; 2 h for T5; and 50 min for T6.

The water entered the clarifier (T4) at mid-elevation and exited at the top of the tank opposite the inflow (fig. 1). To restrict short-circuiting in the clarifier, a baffle was placed in the center of the tank to allow separation of the inflow and outflow.

The aquaculture tank (T5) was designed as a fish refuge to test the potential of incorporating aquaculture in ecological systems treating animal waste. Twelve pounds (5.4 kg) of mixed rocks, gravel, and sand were placed in the bottom of the tank. Woody debris was put in the tank to provide fish habitat. An air diffuser was included to supply sufficient oxygen for the fish survival.

The surface flow wetland (T6) was constructed with an 82 × 35 × 30 cm clear plastic tub. The substrate for the wetland component was clay and organic soils collected from local wetlands. This allowed the system to select suitable microbes and provided a macrophyte seed bank. The depths of the substrate and the water level for T6 were 11 and 8 cm, respectively. Plant specimens were collected from local aquatic ecosystems and nearby research facilities (table 2).

The vegetation in T2 and T3 was placed on nylon netting stretched over a 71 cm diameter metal ring. This ring was

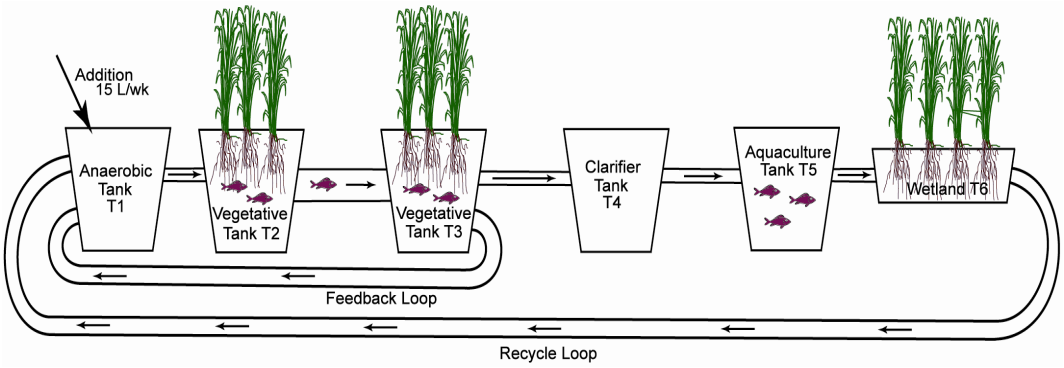

Figure 1. The laboratory-scale ecological treatment system consisted of five 230 L tanks and one 115 L wetland. Water was moved between these tanks and through feedback and recycle loops with airlift pumps. Additional air was pumped into both the aerobic and aquaculture tanks.

Table 2. Vegetation in the wetland component and the two vegetative tanks. The vegetation was added on 17 April 2002.

Aerobic Component (T2)	Aerobic Component (T3)	Wetland Component (T6)
Salix discolor	*Salix discolor*	*Salix discolor*
Salix exigua	*Salix exigua*	*Salix exigua*
Salix eriocephala	*Salix eriocephala*	*Salix eriocephala*
Lemna spp.	*Lemna* spp.	*Lemna* spp.
Colocasia esculenta	*Colocasia esculenta*	*Colocasia esculenta*
Azolla caroliniana	*Utricularia* spp.	*Azzola caroliniana*
Hedychium coronarium	*Cyperus alternifolius*	*Cyperus alternifolius*
Eichornia crassipes	*Elodea canadensis*	*Elodea canadensis*
Iris versicolor	*Iris virsicosa*	*Marsilea* sp.
	Saururus cernuus	*Saururus cernuus*
	Pistia stratiotes	*Pistia stratiotes*
		Canna spp.
		Typha latifolia
		Juncus effusus

supported by six screws that were evenly spaced around the top edge of both tanks. The netting was weighted with 720 g of granite rock to keep it submerged. Air diffusers were added to both vegetative tanks to ensure oxygen saturation. To simulate natural sunlight, one metal halide parabolic reflector lamp (100 W PAR38 flood 3K) was placed over the center of the wetland component and over each vegetative tank. The lamps were controlled by a timer to be activated from 8:00 a.m. to 10:00 p.m. daily.

Species of indigenous fish obtained from local environments and distributors were included to identify suitable species for such a system (table 3). The fish occupied both aerobic tanks and the aquaculture tank. The connection between the two aerobic tanks was large enough that the fish could swim between the tanks, but the connections leaving both tanks were small enough that the fish could not pass. Similarly, the connections into and out of the aquaculture tank prohibited fish passage. All the indigenous fish were added to the system in April 2002. Asian goldfish were added in June 2002.

Table 3. Fish species added to the wetland component and the aerobic tanks. Numbers in parentheses are the number of individuals added.

Common Name	Scientific Name
Aerobic Component (T2)	
Channel catfish (8)	*Ictalurus punctatus*
Silver jaw minnow (2)	*Notropis buccatus*
Stripped shinner (4)	*Luxilus chrysocephalus*
White sucker (1)	*Catostomus commersoni*
Blunt nosed minnow (1)	*Pimephales notatus*
Creek chub (2)	*Semotilus atromaculatus*
Sunfish (1)	*Lepomis macrochirus*
Central stoneroller (24)	*Campostoma* spp.
Goldfish (4)	
Aerobic Component (T3)	
Channel catfish (2)	*Ictalurus punctatus*
Silver jaw minnow (2)	*Notropis buccatus*
Stripped shinner (4)	*Luxilus chrysocephalus*
White sucker (1)	*Catostomus commersoni*
Blunt nosed minnow (1)	*Pimephales notatus*
Creek chub (1)	*Semotilus atromaculatus*
Sunfish (1)	*Lepomis macrochirus*
Central stoneroller (24)	*Campostoma* spp.
Aquaculture Component (T5)	
Silver jaw minnow (3)	*Notropis buccatus*
Stripped shinner (3)	*Luxilus chrysocephalus*
White sucker (1)	*Catostomus commersoni*
Blunt nosed minnow (1)	*Pimephales notatus*
Creek chub (1)	*Semotilus atromaculatus*
Sunfish (1)	*Lepomis macrochirus*
Central stoneroller (24)	*Campostoma* spp.

Data Collection and Laboratory Analysis

Beginning on 24 June 2002, a weekly series of water samples were collected for 12 weeks from T1, T3, T4, and T5 in increments of 3, 24, 96, and 168 h after waste addition. Water samples were not collected from T2 because initial tests indicated little difference in water quality between T2 and T3. The system design did not allow for the collection of an effluent sample from each tank; therefore, all samples were grab samples from 30 cm below the surface of each tank. The waste stream was analyzed before addition to T1 to determine initial concentrations. To differentiate treatment due to biological and physical processes from declines in concentrations due to dilution, the following equation was used to determine concentrations of water quality parameters assuming complete mixing throughout the total working volume of the system:

$$C_d = (W \times C_I / V) \times n \quad (1)$$

where

C_d = diluted concentration (mg L^{-1})
W = volume of waste added (15 L)
C_I = initial concentration (mg L^{-1})
V = total system volume (1136 L)
n = number of waste additions.

The water quality parameters analyzed were: total suspended solids (TSS), total nitrogen (TN), ammonium (NH_4–N), nitrate (NO_3–N), orthophosphate (PO_4–P), total organic carbon (TOC), dissolved organic carbon (DOC), and particulate organic carbon (POC). In addition to water sample collection, a handheld YSI 600R multi-parameter water quality probe with a 650 MDS meter was used to monitor dissolved oxygen (DO), pH, temperature, and conductivity at the time of sample collection in each component of the ecological engineered treatment system.

All water quality analysis was performed by the USDA water quality laboratory housed in the OSU Department of Food, Agricultural, and Biological Engineering and followed standard methods (APHA, 1989). All samples were filtered through a 0.4 μm Millipore nylon membrane filter. The membrane was dried and weighed to determine TSS, and the filtrate was used to determine concentrations of NH_4–N, NO_3–N, and PO_4–P. Block digestions were performed on both unfiltered and filtered samples to allow the calculation of organic nitrogen (ON). Analysis for NH_4–N, NO_3–N, PO_4–P, and TN were analyzed with a Zellweger Analytics Lachat Quickchem FIA+ 8000 Series water quality analyzer, and TOC and DOC were analyzed with a Dorhmann DC-190 total organic carbon analyzer.

Results and Discussion

System Overview

The laboratory-scale system had two significant differences compared to typical Living Machines that limited improvement in water quality. First, because of size constraints, the laboratory-scale system only had half the components of

Table 4. Comparison of the influent and effluent concentrations (mg L^{-1}) of the preliminary system to other ecological treatment systems.

	This Study			Marion, Mass.[a]		Providence, R.I.[b]		Harwich, Mass.[c]		Burlington, Vt.[d]	
	Influent[e]	SE[f]	Effluent[g]	Influent[h]	Effluent	Influent[h]	Effluent	Influent[i]	Effluent	Influent[j]	Effluent[k]
$CBOD_5$	1625.6	135.74	7.05	--	--	--	--	--	--	--	--
BOD_5	--	--	--	--	--	800.00	25.00	104.00	8.00	227	5.9
TSS	1121.05	34.71	3.40	7460	16.00	173.30	5.74	240.00	21.00	189.00	3.00
TN	269.75	4.71	4.24	483.00	6.10	--	--	238.00	9.52	31.00	5.00
TKN	269.75	-	2.28	--	--	--	--	--	--	28.00	2.00
NH_3-N	162.36	2.93	0.02	32.80	0.56	9.40	0.28	--	--	14.00	0.30
NO_3-N	0.00	0.00	2.11	1.10	1.70	--	--	--	--	--	4.00
TP	58.88	1.66	--	51.50	1.50	3.90	2.00	48.00	6.24	6.00	2.00
PO_4-P	40.78	0.62	4.65	--	--	--	--	--	--	--	--
TOC	1279.47	37.62	22.87	--	--	--	--	--	--	--	--
DOC	583.65	17.86	22.35	--	--	--	--	--	--	--	--
POC	695.82	21.26	0.52	--	--	--	--	--	--	--	--

[a] Hamersley et al., 2001.
[b] Todd and Josephson, 1996.
[c] Peterson and Teal, 1996.
[d] Austin, 2000.
[e] Average (N = 23) of dairy wastewater grab samples without screening or degritting.
[f] Standard error for system influent (N = 23).
[g] Average (N = 6) of grab samples from each tank at 168 h.
[h] Screened and degritted septage.
[i] Screened and degritted septage, primary treatment supernatant.
[j] Raw sewage.
[k] Flow weighted average means of two treatment lines.

standard Living Machines. Second, because this was a closed system (i.e., no effluent), there was a risk that PO_4-P and other conservative elements would increase over time. Results from T6, the wetland mesocosm, have been excluded from this study due to inconsistencies in the data. The design of this component made sampling difficult, and sample contamination was unavoidable.

Influent wastewater concentrations averaged (n = 23) 1121.05 mg L^{-1} for TSS, 269.75 mg L^{-1} for TN, 269.75 mg L^{-1} for TKN, 162.36 mg L^{-1} for NH_4-N, 0.00 mg L^{-1} for NO_3-N, 40.78 mg L^{-1} for PO_4-P, 1279.47 mg L^{-1} for TOC, and 583.65 mg L^{-1} for DOC (table 4).

Reductions were seen in each tank for all parameters after 168 h (table 5). While final concentrations of NO_3-N exceeded initial concentrations, there was a reduction of NO_3-N in comparing final concentrations to concentrations at 3 h (table 5). The nitrification of NH_4-N increased the concentrations of NO_3 from the initial near 0 mg L^{-1} to below 3 mg L^{-1}. Since the design of this system did not allow for an effluent, the water was continually recycled through the system. Therefore, despite differences between tank environments, there were no significant differences when final concentrations at 168 h were compared between the different tanks. The one exception to this trend was TSS in T3 (table 5).

Total Suspended Solids

The greatest reduction of TSS was observed at 96 h, when all tanks had greater than 95% reductions (table 5). While all the tanks had a high removal of solids, the concentration remaining in the vegetative tank (T3) was at least 10 to 20 times greater than the concentrations remaining in all other tanks.

Remaining solids not digested in T1 either accumulated on the lava rocks or were pumped to the vegetative tanks for further treatment. Agitation of the vegetative tanks with air diffusing stones prohibited optimal settling and contributed to the elevated solid concentrations relative to the other tanks. Similar results were reported by Hamersley et al. (2001) in a study of a solar aquatic system to treat septage. They observed that during anaerobic digestion, solids were susceptible to accumulation and microbial degradation, while in the aerobic environment accumulation did not occur due to continual aeration.

Nitrogen

The following discussion is focused on the results from T1 because N concentrations were similar between all tanks. Forms of N examined included total nitrogen (TN), total Kjeldahl nitrogen (TKN), ammonium (NH_4-N), and nitrate + nitrite (NO_3-N). There was a rapid reduction of NH_4-N in the first 24 h (98.26% reduction in T1). The primary means of removal included volatilization, vegetative uptake, and conversion to NO_3-N by nitrification. Volatilization is dependent on pH and was estimated based on the following equation (Rao et al., 1984):

$$C_V = 5.8 \times 10^{(pH - 10)} \times C_W \quad (2)$$

where

C_V = volatilized NH_3-N
C_W = NH_4-N in water.

The transformation of NH_4-N to NH_3-N occurs rapidly at pH values greater than 8. After NH_3-N is formed, volatilization follows (Rao et al., 1984; Martin and Reddy, 1997; Mitsch and Gosselink, 2000). Through the application of the above equation, volatilization accounted for only 1% to 2% (0.51 mg L^{-1}) of the NH_4-N transformation in the laboratory-scale system. The pH of the water averaged 6.83 in the anaerobic tank, but was at or above 8 in the other tanks (table 6). The remaining NH_4-N was assumed to be converted to NO_3-N by nitrification or assimilated by the vegetation. This concentration (25.21 mg L^{-1}) accounted for 98% of the initial NH_4-N concentration. Since nutrient analysis of vegetation was not performed, the percentage of NH_4-N reduction due to vegetative uptake or nitrification could not be determined.

Table 5. Water samples were collected over a 12-week period and analyzed for TSS, TN, TKN, NH_4-N, NO_3-N, PO_4-P, TOC, DOC, and POC. All data points are reported as an average of the 12 weeks. Negative numbers indicate an increase in concentrations. All 0 h concentrations are based on diluted initial concentration assuming complete mixing throughout the total system volume.

		Tank T1			Tank T3			Tank T4			Tank T5		
	Time (h)	Conc. (mg L^{-1})	SE	Percent Reduct.	Conc. (mg L^{-1})	SE	Percent Reduct.	Conc. (mg L^{-1})	SE	Percent Reduct.	Conc. (mg L^{-1})	SE	Percent Reduct.
TSS	0	700.52	--	0.00	700.52	--	0.00	700.52	--	0.00	700.52	--	0.00
	3	47.35	6.18	93.24	47.88	1.42	93.16	8.29	0.70	98.82	3.98	0.32	99.43
	24	10.15	0.99	98.55	35.77	2.63	94.89	8.04	0.54	98.85	8.28	0.60	98.82
	96	1.47	0.13	99.79	28.83	1.92	95.88	1.30	0.10	99.81	3.28	0.23	99.53
	168	4.13	0.83	**99.41**	41.91	3.55	**94.02**	1.66	0.12	**99.76**	4.42	0.47	**99.37**
TN	0	42.74	--	0.00	42.74	--	0.00	42.74	--	0.00	42.74	--	0.00
	3	5.85	0.26	86.32	5.78	0.09	86.47	5.10	0.10	88.07	4.47	0.07	89.54
	24	5.56	0.12	86.99	6.08	0.12	85.77	5.48	0.10	87.19	5.40	0.08	87.36
	96	4.76	0.06	88.86	5.28	0.08	87.64	5.17	0.08	87.91	4.89	0.07	88.57
	168	4.25	0.06	**90.05**	4.64	0.08	**89.14**	4.03	0.09	**90.57**	4.04	0.06	**90.55**
TKN	0	42.74	--	0.00	42.74	--	0.00	42.74	--	0.00	42.74	--	0.00
	3	3.18	0.16	92.55	3.19	0.03	92.55	2.32	0.03	94.57	2.19	0.03	94.88
	24	2.66	0.10	93.77	2.92	0.05	93.16	2.41	0.01	94.37	2.47	0.04	94.23
	96	2.21	0.03	94.83	2.66	0.05	93.77	2.19	0.01	94.88	2.29	0.04	94.64
	168	2.27	0.03	**94.68**	2.72	0.04	**93.64**	2.06	0.02	**95.17**	2.05	0.02	**95.20**
NH_4-N	0	25.72	--	0.00	25.72	--	0.00	25.72	--	0.00	25.72	--	0.00
	3	2.07	0.17	91.93	1.92	0.12	92.55	0.47	0.05	98.18	0.05	0.01	99.80
	24	0.45	0.05	98.26	0.40	0.06	98.46	0.60	0.07	97.65	0.52	0.06	97.98
	96	0.02	0.00	99.93	0.02	0.00	99.92	0.01	0.00	99.95	0.01	0.00	99.94
	168	0.02	0.00	**99.94**	0.02	0.00	**99.94**	0.01	0.00	**99.98**	0.02	0.00	**99.92**
NO_3-N[a]	0	0.00	--	--	0.00	--	--	0.00	--	--	0.00	--	--
	3	2.91	0.17	--	2.79	0.07	--	2.89	0.06	--	2.23	0.05	--
	24	2.93	0.05	-0.63	3.20	0.07	-14.84	3.11	0.08	-7.49	2.98	0.05	-34.06
	96	2.46	0.06	15.64	2.49	0.06	22.19	2.82	0.07	9.11	2.49	0.06	16.48
	168	2.07	0.05	**28.97**	2.17	0.06	**32.34**	2.10	0.06	**32.23**	2.10	0.06	**29.73**
PO_4-P	0	6.48	--	0.00	6.48	--	0.00	6.48	--	0.00	6.48	--	0.00
	3	4.91	0.14	24.16	4.81	0.07	25.76	4.76	0.07	26.57	4.60	0.06	29.00
	24	4.88	0.06	24.63	4.89	0.06	24.57	5.02	0.06	22.57	4.96	0.07	23.39
	96	4.96	0.07	23.52	4.94	0.06	23.77	4.74	0.08	26.83	4.96	0.06	23.49
	168	4.66	0.06	**28.16**	4.49	0.09	**30.68**	4.77	0.06	**26.34**	4.70	0.06	**27.53**
TOC	0	202.73	--	0.00	202.73	--	0.00	202.73	--	0.00	202.73	--	0.00
	3	28.24	0.92	86.07	28.69	0.30	85.85	23.25	0.13	88.53	22.93	0.13	88.69
	24	23.91	0.10	88.21	24.91	0.21	87.71	23.59	0.12	88.36	24.11	0.16	88.11
	96	23.32	0.11	88.50	23.62	0.20	88.35	22.11	0.15	89.10	23.43	0.11	88.44
	168	22.75	0.15	88.78	23.64	0.17	88.34	22.20	0.10	89.05	22.89	0.16	88.71
DOC	0	92.32	--	0.00	92.32	--	0.00	92.32	--	0.00	92.32	--	0.00
	3	23.47	0.62	74.58	22.78	0.17	75.33	22.27	0.17	75.87	21.82	0.17	76.36
	24	23.28	0.21	74.78	22.66	0.17	75.45	22.90	0.18	75.20	22.98	0.19	75.11
	96	22.96	0.21	75.13	22.86	0.22	75.24	22.58	0.17	75.54	23.13	0.14	74.95
	168	22.62	0.10	**75.49**	22.37	0.12	**75.77**	22.70	0.08	**75.41**	21.72	0.32	**76.47**
POC[b]	0	110.41	--	0.00	110.41	--	0.00	110.41	--	0.00	110.41	--	0.00
	3	4.77	--	95.68	5.91	--	94.65	0.98	--	99.11	1.11	--	98.99
	24	0.63	--	99.43	2.25	--	97.96	0.69	--	99.38	1.13	--	98.98
	96	0.36	--	99.67	0.76	--	99.31	-0.47	--	100.43	0.3	--	99.73
	168	0.13	--	**99.88**	1.27	--	**98.85**	-0.5	--	**100.45**	1.17	--	**98.94**

[a] Percent reduction was calculated from the 3 h value, not the 0 h value.
[b] Calculated values.

Past studies of aquatic macrophytes and N reduction have demonstrated that vegetative assimilation usually accounts for a small percentage of N removal (Bachand and Horne, 2000; Brix and Schierup, 1989). Therefore, it was likely that most of the NH_4-N reduction was due to nitrification. Biofilms on plant roots, suspended organic material, and tank walls all contain nitrifying bacteria that enhance nitrification (Hamersley et al., 2001). In our system, there was an abundance of biofilms to support high nitrification rates during the first 24 h of treatment. Nitrification was greatly reduced after the first 24 h because 98% of the NH_4-N was removed within the first 24 h. Hamersley et al. (2001), using a solar aquatic system, similarly concluded that as NH_4-N concentrations decreased, so did the nitrification rates in the system.

A portion of the total NH_4-N that was reduced is accounted for in the residual NO_3-N concentration (2.07 mg L^{-1}) at 168 h. The relationship between the transformations of NH_4-N to NO_3-N to the overall N reduction is illustrated

Table 6. A YSI instrument was used to collect temperature, dissolved oxygen (DO), and pH data for each tank. All data points are an average for last 9 weeks of the study period.

Tank	Time (h)	Temperature (°C)		% DO		DO (mg L^{-1})		pH	
		Average	SE	Average	SE	Average	SE	Average	SE
T1	3	21.95	0.13	5.21	1.24	0.45	0.11	6.74	0.02
	24	22.11	0.24	2.00	0.40	0.17	0.03	6.87	0.02
	96	21.78	0.15	2.61	0.42	0.22	0.04	6.85	0.02
	168	21.60	0.16	2.80	0.70	0.24	0.06	6.86	0.05
T2	3	21.97	0.14	106.69	0.66	9.00	0.06	8.05	0.01
	24	22.23	0.28	92.62	1.01	7.75	0.10	8.04	0.00
	96	21.76	0.15	101.16	0.93	8.55	0.09	7.96	0.01
	168	21.52	0.15	102.18	1.95	8.66	0.15	7.94	0.03
T3	3	21.99	0.14	109.98	0.55	9.27	0.05	8.27	0.03
	24	22.25	0.28	99.37	0.94	8.32	0.11	8.09	0.01
	96	21.79	0.15	104.54	0.87	8.84	0.09	8.04	0.01
	168	21.54	0.15	105.53	1.79	8.94	0.14	8.12	0.02
T4	3	22.01	0.14	104.33	0.51	8.82	0.05	8.13	0.02
	24	22.24	0.28	98.05	1.41	8.22	0.13	8.10	0.01
	96	21.81	0.15	93.56	1.63	7.91	0.15	8.01	0.02
	168	21.57	0.16	100.35	1.61	8.50	0.13	8.11	0.02
T5	3	21.88	0.14	107.51	0.57	9.08	0.06	8.17	0.01
	24	22.07	0.29	105.53	1.28	8.88	0.12	8.14	0.01
	96	21.65	0.15	101.87	0.80	8.63	0.08	8.06	0.01
	168	21.42	0.15	103.45	1.69	8.78	0.13	8.15	0.02

by comparing the initial and final percentages of TN accounted for by these two N species (fig. 2). Initially, the NH_4-N concentration (25.72 mg L^{-1}) accounted for 60.2% and NO_3-N accounted for 0% of the TN in the water. After 168 h, the concentration of NO_3-N (2.07 mg L^{-1}) accounted for 50% of the TN remaining in the water, while NH_4-N accounted for only 0.1%. The other 50% of TN at 168 h consisted mostly of ON (TN_{final} - NO_3 - N_{final} - NH_4-N_{final} = ON_{final}, table 5). Overall, the combined pathway of nitrification-denitrification and vegetative uptake of NO_3-N and NH_4-N accounted for 57% of the N reduction (24.29/42.74 mg L^{-1}). The removal of ON ($ON_{initial}$ - ON_{final}) was responsible for 5.1% of the TN removed from the system (fig. 2).

Following these assumptions and calculations, we cannot account for 34.7% of the TN. Sedimentation and mineralization undoubtedly represented a portion of this unquantified N. Previous studies found that saturated water conditions can limit mineralization rates (Oomes et al., 1997); therefore, sedimentation may represent more of the unaccounted N as compared to mineralization.

PHOSPHORUS

The system produced lower rates of PO_4-P removal compared to similar systems. System designs that included preliminary solids removal and primary treatment before ecological treatment have produced higher P removal (Hamersley et al., 2001; Peterson and Teal, 1996) (table 4). Similar

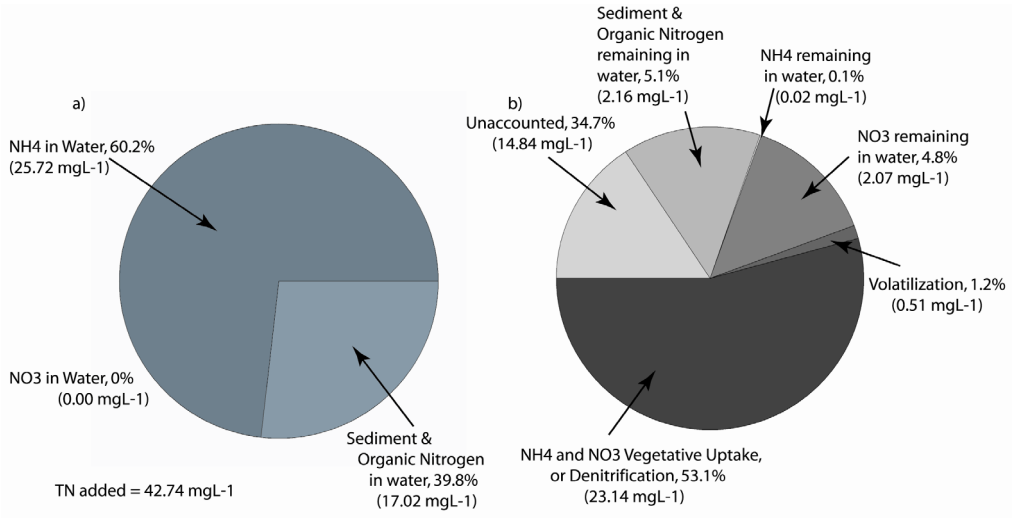

Figure 2. Influent fractions of nitrogen in the wastewater added to the system: (a) nitrogen transformations and removal from the system at 168 h expressed as percentages of total initial nitrogen added, and (b) all percentages for influent nitrogen, calculated from the diluted influent concentrations (table 4). All percentages for nitrogen transformation and removal were calculated from the 168 h data of T1 (table 5) and the equations and estimations stated.

to systems without preliminary solids removal or primary treatment (Todd and Josephson, 1996; Austin, 2000) (table 4), our system also produced lower removal rates for P. On average, it removed 28% of the PO_4-P added to the system, as compared to TP removals of 97% (Hamersley et al., 2001), 49% (Todd and Josephson, 1996), 87% (Peterson and Teal, 1996), and 67% (Austin, 2000) (table 4) for similar systems. Since these systems were analyzed for TP, not PO_4-P, it is hard to compare previous system results to the results for this system. If water quality analysis of TP were available for this study, a greater reduction might have been observed. However, PO_4-P data are all that were available.

Due to poor removal, PO_4-P accumulated in the system over the 12-week study period (fig. 3e). All other variables decreased or reached a steady concentration. The primary processes involved in the removal of P are sedimentation, adsorption, and vegetative uptake (Braskerud, 2002). Most P in the water column is incorporated into the sediment by soil adsorption (Mitsch and Gosselink, 2000). The biological component of this cycle consists of plant uptake of the dissolved phosphates with successive transfer through the food chain (Schmitz, 1996). Since the soil substrate, vegetation, and sedimentation were not analyzed in this study, we cannot determine their contribution to PO_4-P removal. However, Peterson and Teal (1996), using a solar aquatic system, found that vegetation removed only 3% of the TP added, while 65% was removed through sedimentation. For our system, a large portion of the solids (99%) was removed from the water column in the first 24 h (table 5). This removal was through sedimentation, soil adsorption, or anaerobic/aerobic digestion. However, digestion of solids by microbes could result in a release of PO_4-P from the solids.

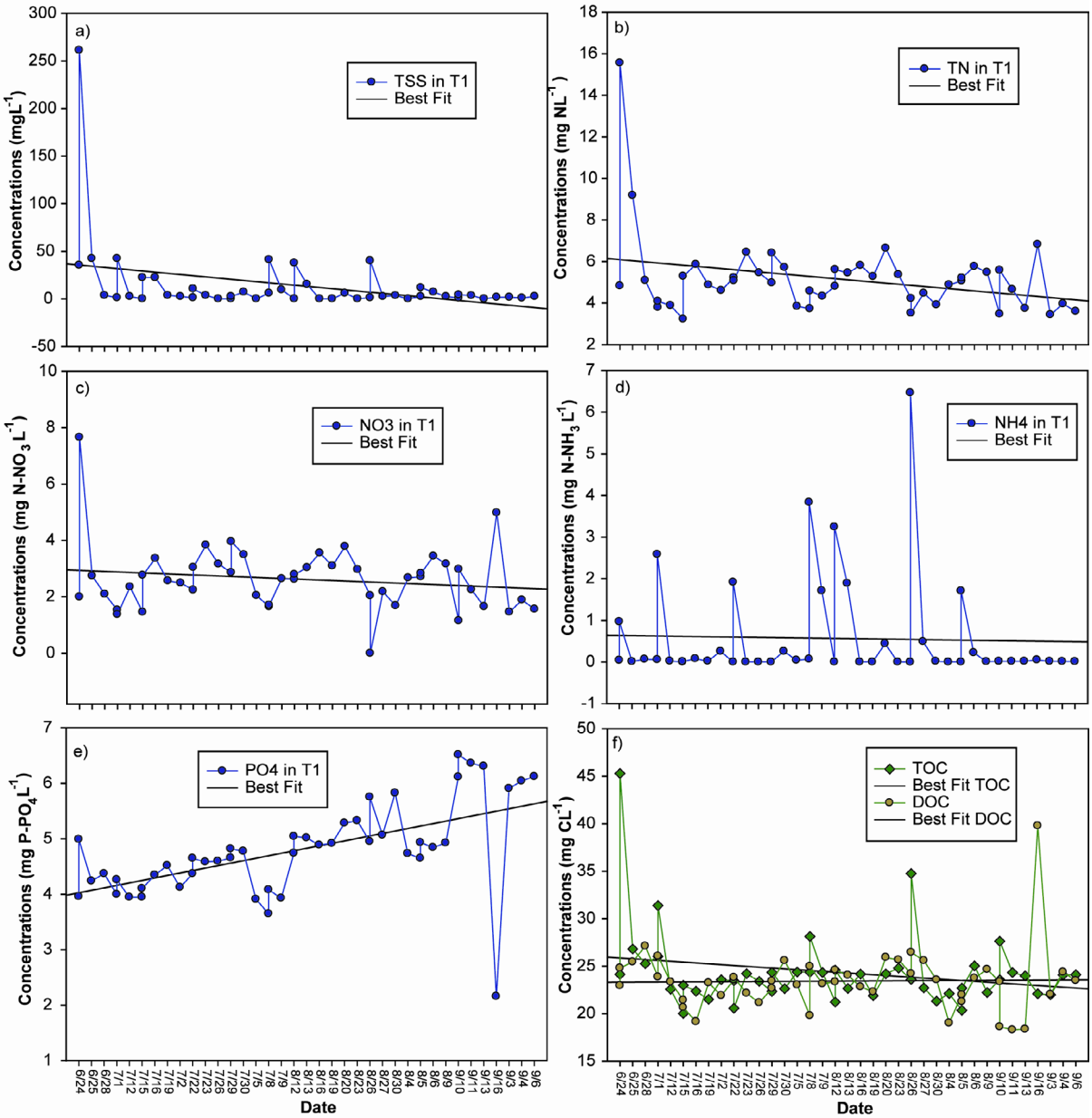

Figure 3. Observed trends for each parameter in tank 1 over the 12-week study period: (a) total suspended solids (TSS), (b) total nitrogen (TN), (c) nitrite + nitrate (NO_3-N), (d) ammonium (NH_4-N), (e) orthophosphate (PO_4-P), and (f) total organic carbon (TOC) and dissolved organic carbon (DOC).

The substrate used and the size of the wetland component also affected the P retention capabilities of the system. The organic substrate used in the wetland component did not maximize exchange sites for the binding of PO_4–P. After the initial available sites were exhausted, poor removal of PO_4–P followed. Past studies indicated that different substrates, such as limestone gravel, should be considered in future designs to increase P removal. In a Living Machine for the treatment of municipal sewage, a gravel bed wetland component with an area of 2.2 m^2 resulted in P reductions near 50% (Todd and Josephson, 1996). A solar aquatic system utilizing a gravel bed wetland component with an area of 11.5 m^2 resulted in an 87% reduction in TP in the treatment of domestic and commercial raw septage (Peterson and Teal, 1996). Incorporating a gravel bed wetland into future designs may increase the potential for P removal.

Organic Carbon

Much of the TOC present in the system was in dissolved form. In T1 at 3 h after waste addition, 83% of the TOC (28.24 mg C L^{-1}) was accounted for as DOC (23.47 mg C L^{-1}) (table 5). However, at 168 h, the concentrations of TOC and DOC were equivalent. This indicated that a substantial amount of the particulate organic carbon (POC) was removed from the system either through sedimentation or biological decomposition. Initial and final concentrations for POC were 110.4 and 0.13 mg C L^{-1} for T1, respectively, resulting in a reduction of 99.8% (table 5). However, the removal of TOC and DOC was lower, 88.8% and 75.5%, respectively. Through biological decomposition, a portion of the POC was converted to DOC, which hindered the DOC removal (Moran and Hodson, 1994). While this degradation process provides an available carbon source to drive denitrification (Pinney et al., 2000), it also releases humic substances into the bulk DOC pool (Moran and Hudson, 1994) that are not available for microbial consumption (Nguyen, 2000). For this system, the available carbon was exhausted quickly, leaving behind recalcitrant forms of DOC. This limited denitrification and resulted in residual concentrations of NO_3–N in the system (table 5). Future studies may need to include the addition of an available carbon source to further enhance the denitrification process (Austin, 2000; Hamersley and Howe, 2003). Another means of increasing the available carbon would be by relocating the feedback loop from the clarifier to the anaerobic tank. This would allow recycling of activated biosolids necessary for waste treatment (Hamersley et al., 2001).

Value-Added Products

Because the primary objective of this study was to quantify water quality improvements, only qualitative observations were used to assess vegetative and fish biomass. A high mortality rate was observed for the indigenous Midwestern fish species that were initially added to the system. The potential pathogen source in the wastewater could have contributed to this loss in indigenous fish. The goldfish that were later added to the system thrived. Over a 9-week period, the goldfish increased in biomass by 123%, from an average of 6.2 to 13.8 g per fish.

It was also observed that the herbaceous vegetation had high rates of growth in the wetland component and the vegetative tanks. The woody vegetation, however, declined in biomass. Both vegetative tanks and the wetland were planted with three willow species (table 2) and observed for growth. Initially, all the willows appeared to gain biomass. However, three weeks into the study, T2 and T3 lost one willow each. By the end of the study period, four of the nine willows had died (T1 = 1, T2 = 2, and T3 = 1), and the other five showed signs of stress. The laboratory constraints on natural sunlight could have contributed to this observed difference between the woody and the herbaceous vegetation. Future studies of vegetation and fish will include a quantitative analysis of biomass changes as well as nutrient assimilation.

Conclusion

Our results indicate the potential for this technology to be effective in the treatment of highly concentrated dairy wastewater. Promising reduction rates have been achieved for suspended solids, TOC, TN, NH_4^+, and TP (99.8%, 19.4%, 46.3%, 99.8%, and 28.2%, respectively). Anticipated improvements for future system designs and studies should include the following:

Increase quantity of waste additions. The results of the preliminary system demonstrated the potential for the addition and treatment of greater quantities of wastewater. Due to the laboratory-scale limitations, this preliminary study treated small amounts of wastewater. In order for this application to be useful, much larger quantities of wastewater need to be treated. By increasing the quantity of wastewater treated, we may need to address other concerns that were not addressed in this study, such as the use of odor control devices.

Increasing the size of the wetland component. An increase in the size of the wetland component should improve the removal rates for PO_3^- due to an increase in available substrate surface area for ion exchange. An increase in size should improvement water quality parameters as well as allow for the treatment of larger quantities of wastewater.

Consideration of the wetland substrate. Materials with greater anion exchange capacity (positively charged surface) have a greater affinity for P ions. Specially prepared clays, iron and aluminum rich materials, and limestone can enhance P storage. The storage capacity of surface flow wetlands can be exhausted quickly, while subsurface flow wetlands can be designed to have large P storage via adsorption. By altering the flow and design of the wetland component to a subsurface flow instead of a surface flow wetland, the treatment capacity could be improved.

Change location of feedback loop. The feedback loop from the vegetative tank should be moved to the settling tank to deliver solids and water to the anaerobic tank. This will recycle activated sludge to the anaerobic tank and, in turn, potentially increase denitrification rates in the anaerobic tank and reduce solids collected in the clarifier.

In-depth analysis of the system. While the results of the preliminary study are useful, an analysis of the pathogens, vegetation, fish biomass, sedimentation, and denitrification are also needed. A complete analysis will not only assist in determining the pertinent biological processes involved in the treatment of highly concentrated dairy wastewater, but also aid in the future designs of ecologically engineered treatment systems.

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REFERENCES

Adey, W. H., C. Luckett, and M. Smith. 1996. Purification of industrially contaminated groundwaters using controlled ecosystems. *Ecological Eng*. 7(3): 191-212.

APHA. 1989. *Standard Methods for the Examination of Water and Wastewater.* 17th ed. Washington D.C.: American Public Heath Association.

Austin, D. 2000. Final report on the South Burlington, Vermont, advanced ecological engineered system (AEES) for waste treatment. Washington D.C.: U.S. Environmental Protection Agency.

Bachand, P. A. M., and A. J. Horne. 2000. Denitrification in constructed free-water surface wetlands: II. Effects of vegetation and temperature. *Ecological Eng.* 14(1-2): 17-32.

Braskerud, B. C. 2002. Factors affecting phosphorus retention in small constructed wetlands treating agricultural nonpoint-source pollution. *Ecological Eng*. 19(1): 41-61.

Brix, H., and H. H. Schierup. 1989. The use of aquatic macrophytes in water-pollution control. *Ambio* 18(2): 100-107.

Canovas, S., B. Picot, and C. Casellas. 1996. Seasonal development of phytoplankton and zooplankton in high-rate algal ponds. *Water Sci. and Tech.* 33(7): 199-206.

Costa-Pierce, B. A. 1998. Preliminary investigation of an integrated aquaculture-wetland ecosystem using tertiary-treated municipal wastewater in Los Angeles County, California. *Ecological Eng*. 10(4): 341-354.

Craggs, R. J., W. H. Adey, and K. R. Jenson. 1996a. Phosphorus removal from wastewater using an algal turf scrubber. *Water Sci. and Tech.* 33(7): 191-198.

Craggs, R. J., V. J. Smith, and P. J. Mcauley. 1996b. A controlled stream mesocosm for tertiary treatment of sewage. *Ecological Eng*. 6(1-3): 149-469.

Drenner, R. W., D. J. Day, S. J. Basham, D. Smith, and S. I. Jensen. 1997. Ecological water treatment system for removal of phosphorus and nitrogen from polluted water. *Ecological Application* 7(2): 381-390.

Geary, P. M., and J. A. Moore. 1999. Suitability of a treatment wetland for dairy wastewaters. *Water Sci. and Tech.* 40(3): 179-185.

Hamersley, M. R. and B. L. Howe. 2003. Contribution of denitrification to nitrogen, carbon, and oxygen cycling in tidal creek sediments of a New England salt marsh. *Marine Ecology – Progress Series* 262: 55-69.

Hamersley, M. R., B. L. Howes, D. S. White, S. Johnke, D. Young, S. B. Peterson, and J. M. Teal. 2001. Nitrogen balance and cycling in an ecologically engineered septage treatment system. *Ecological Eng*. 18(1): 61-75.

Martin, J. F., and K. R. Reddy. 1997. Interaction and spatial distribution of wetland nitrogen processes. *Ecological Modelling* 105(1): 1-21.

Metcalf and Eddy. 2003. *Wastewater Engineering: Treatment and Reuse.* 4th ed. Boston, Mass: McGraw Hill.

Mitsch, W. J., and J. G. Gosselink. 2000. *Wetlands.* New York, N.Y.: John Wiley and Sons.

Moran, M. A., and R. E. Hodson. 1994. Dissolved humic substances of vascular plant origin in a coastal marine environment. *Limnol. Oceanogr.* 39(4): 762-771.

Newman, J. M., J. C. Clausen, and J. A. Neafsey. 2000. Seasonal performances of a wetland constructed to process dairy milkhouse wastewater in Connecticut. *Ecological Eng*. 14(1-2): 181-198.

Nguyen, L. M. 2000. Organic matter composition, microbial biomass, and microbial activity in gravel-bed constructed wetlands treating farm dairy wastewaters. *Ecological Eng*. 16(2): 199-221.

Oomes, M. J. M., P. J. Kuikman, and F. H. H. Jacobs. 1997. Nitrogen availability and uptake by grassland in mesocosms at two water levels and water qualities. *Plant and Soil* 192(2): 249-259.

Peterson, S. B., and J. M. Teal. 1996. The role of plants in ecological engineered wastewater treatment systems. *Ecological Eng*. 6(1-3): 137-148.

Pinney, M. L., P. K. Westerhoff, and L. Baker. 2000. Transformations in dissolved organic carbon through constructed wetlands. *Water Res*. 34(6): 1897-1911.

Rao, P. S. C., R. E. Jessup, and K. R. Reddy. 1984. Simulation of nitrogen dynamics in flooded soils. *Soil Sci.* 138(1): 54-62.

Rose, P. D., B. A. Maart, K. M. Dunn, R. A. Rowswell, and P. Britz. 1996. High rate algal oxidation ponding for the treatment of tannery effluents. *Water Sci. and Tech.* 33(7): 219-227.

Schaafsma, J. A., A. H. Baldwin, and C. A. Streb. 2000. An evaluation of s constructed wetland to treat wastewater from a dairy farm in Maryland, USA. *Ecological Eng*. 14(1-2): 199-206.

Schmitz, R. J. 1996. *Introduction to Water Pollution Biology.* Houston, Tex.: Gulf Publishing.

Schroeder, G. L., G. Wohlfarth, A. Alkon, A. Halevy, and H. Krueger. 1990. The dominance of algal-based food webs in fish ponds receiving chemical fertilizers plus organic manures. *Aquaculture* 86(2-3): 219-229.

Sladeckova, A. 1994. Green-algae and waste treatment technology. *Biologia*. 49(4): 615-619.

Smith, D. W. 1993. Wastewater treatment with complementary filter feeders: A new method to control excessive suspended solids and nutrient stabilization ponds. *Water Environ. Res.* 65(5): 650-654.

Spataru, P., G. W. Wohlfarth, and G. Hulata. 1983. Studies on the natural food of different fish species in intensively manured polyculture ponds. *Aquaculture* 35: 283-298.

Tanner, C. C., J. S. Clayton, and M. P. Upsdell. 1995. Effects of loading rate and planting on treatment of dairy farm wastewaters in constructed wetlands: II. Removal of nitrogen and phosphorus. *Water Resources* 29(1): 27-34.

Todd, J., and B. Josephson. 1996. The design of living technologies for waste treatment. *Ecological Eng*. 6(1-3): 109-136.

Todd, J., E. J. G. Brown, and E. Wells. 2003. Ecological design applied. *Ecological Eng*. 20(5): 421-440.

Vymazal, J. 1988. The use of periphyton communities for nutrient removal from polluted streams. *Hydrobiologia* 166(3): 225-237.

Wohlfarth, G. W., and G. L. Schroeder. 1979. Use of manure in fish farming: A review. *Agric. Waste* 1(4): 279-299.

Wohlfarth G. W., R. Moav, and G. Hulata. 1983. A genotype-environment interaction for growth rate in the common carp, growing in intensively manured ponds. *Aquaculture* 33: 187-195.

Wohlfarth, G. W., G. Hulata, I. Karplus, and A. Halevy. 1985. Polyculture of freshwater prawn *Machrobrachium rosenbergii* in intensively manured ponds, and the effect of stocking rate of prawns and the fish on their production characteristics. *Aquaculture* 46(2): 143-156.